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Richard A. Schoettger, Ph.D.

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IMPACT OF PERMETHRIN ON ZOOPLANKTON IN HIGH LATITUDE EXPERIMENTAL PONDS

by

Thomas W. La Point[1], O.D. Zhavoronkova[2] and D.F. Pavlov[2]

ABSTRACT

This study evaluated the effect of a range of concentrations (0.5-50 ug liter^{-1}) of permethrin on experimental pond zooplankton in northern USSR. Each pond was dosed once and the experiment was unreplicated. Samples were collected from each pond prior to, 7 days and 11 days after sampling. High sampling variances precluded measuring statistically significant differences among the ponds, with the exception of rotifers. Certain rotifer species appear to be highly sensitive to pyrethroid insecticides. The value of this descriptive study is in outlining future studies of the response of planktonic organisms to one-time or pulsed dosing of chemicals in experimental ecosystems. The frequency and duration of exposure needs to be taken into account relative to the generation time and reproductive capacity of the organisms under study.

INTRODUCTION

A critical consideration in measuring the effect of pesticides on non-target aquatic species is integrating exposure and effect, relative to life history characteristics of the exposed organisms. Trophic status, including nutrient concentration, suspended particulate, and

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[1]U.S. Fish and Wildlife Service, National Fisheries Contaminant Research Center, Columbia, MO (USA) *Present address:* Department of Environmental Toxicology and Institute of Wildlife and Environmental Toxicology (TIWET), Clemson University, Pendleton, SC 29670 (USA).
[2]Institute of Biology of Inland Waters, Russian Academy of Sciences, Borok, Jaroslavl Region, Russia

dissolved and particulate organic matter affects chemical bioavailability in surface waters (Kaiser 1980, McCarthy & Bartell 1988). Morpho-edaphic components of the environment (depth, slope, current, physical-chemical nature of the substrate) also exert a strong influence on the nature of the biota in aquatic environments (Minshall 1988, Johnson *et al.* 1993). Biotic response to a contaminant depends upon life history characteristics of the individual species (body size, reproductive potential, and habitat and food requirements) (Barnthouse *et al.* 1987, Barnthouse *et al.* 1990, La Point 1994). Further, consideration must be made for conservatism, as little is known of inter-specific interactions in planktonic food webs. Most of the work to date has dealt with effects on zooplankton (Liber *et al.* 1994, DeNoyelles *et al.* 1994), benthos (Fairchild *et al.* 1992, Lozano *et al.* 1992, Giddings *et al.* 1994) and fishes (Fairchild *et al.* 1994, Barnthouse *et al.* 1987). Planktonic and benthic fauna are important in that they form the base of food webs in natural ecosystems and are chiefly responsible for decomposition of particulate and in influencing the overall secondary production within aquatic ecosystems. How these not-target groups respond to pesticides will influence rates of decomposition, primary production, nutrient cycling, and effects on organisms higher in the food chain (Dewey and DeNoyelles 1994).

Increasingly, more information is being collected on micro fauna, particularly the zooplankton. However, one group of micro invertebrates for which there is very little information is that of water mites, *Hydracarina*. *Hydracarina* are widely distributed all over the world, except the Antarctic region, and inhabit almost all types of water bodies (Gledhill 1985). The population density of water mites may reach extremely high levels (Tuzovshiy 1972). The nymphs and imagoes of water mites are mainly predators and play a great role in the functioning of aquatic ecosystems (Ten Winkel and Davids 1985, Kutikova 1977). At the same time, this group of Chelicerata is comparatively poorly studied in most aspects of aquatic ecology and particularly in aquatic ecotoxicology. The number of papers or reports dealing with the influence of various toxic substances on water mites is comparatively small and the conclusions made in these papers are often contradictory (Alexeev 1971, Nair *et al.* 1977, Stephenson 1982). For example, V.A. Alexeev (1986) stated that *Hydracarina* are generally resistant to a variety of toxicants. On the other hand, it has been shown that they are very sensitive to toxic action of pesticides and may be useful for the purposes of bioindication (Nair *et al.* 1977).

Synthetic pyrethroids are becoming widely used in agricultural regions across the world. This class of insecticide is characterized by low water solubility, high lipophilicity, high toxicity to fish and invertebrates, and relatively low toxicity to mammals. Because synthetic pyrethroids are widely used within the USA and Russia, it is important that the effect of these chemicals on non-target aquatic species and ecosystems be studied. The research described in this paper was conducted at the experimental ponds of the Institute of Biology of Inland Waters (IBIW), Russian Academy of Sciences, Borok, Jaroslavl Region, Russia. Permethrin (3-phenoxybenzyl (1RS)-cis, trans-3-(2,2-dichlorovinyl)-2,2-dimethylcyclopropanecarboxylate) was used as the commercial product Pounce, manufactured by FMC, Inc. Pounce has 38.47% active ingredient, 50.7% xylene, and 10.8% related reaction and inert products.

MATERIALS AND METHODS

DESCRIPTION OF STUDY PONDS

The IBIW is located on the Rybynsk Reservoir in the Jaroslavl Region of Russia, roughly 330 km northeast of Moscow at 58 degrees N latitude. The experimental ponds at the IBIW have a surface area of 800 m^2, volume of 400 m^3, and average depth of 50 cm. The ponds are characteristic of high-latitude, oligotrophic ponds (Table 1).

Five ponds were used in this experiment: control, acetone control, and three ponds treated with permethrin to achieve concentrations of 0.5, 5.0 and 50.0 ug $liter^{-1}$ (pond numbers 1, 2, 3, 4, and 5, respectively). Acetone was used as the carrier for permethrin; 1 liter was dosed into each pond with the permethrin by sub-surface application (at 15 to 20 cm depth) using a hand-held sprayer. The concentration of permethrin in 1-liter acetone was 0.52 ml, 5.2 ml, and 52.0 ml to achieve the 0.5, 5.0 and 50.0 ug $liter^{-1}$ concentrations, respectively. The chemical was mixed into the ponds by transects taken across the surface in a boat. Ponds were treated once, on May 25, 1989.

RESIDUE ANALYSIS

Permethrin samples were intended to be analyzed by gas chromatography following collection and transport to the USA. Permethrin samples were collected on days 1 and 11 following dosing. Collected water samples were extracted onto C-18 packed glass columns, frozen, and carried to the National Fisheries Contaminant Research Center in Columbia, Missouri, for analysis. Unfortunately, the samples were not allowed into the country; hence, results are in terms of nominal permethrin concentrations.

Table 1. Water chemistry of Russian Academy of Sciences Institute (IBIW) ponds at Sunuga, Borok, USSR.

Pond No.	Control	Acetone Control	0.5 µg/l	5.0 µg/l	50.0 µg/l
Total suspended particulates, mg/l	1.0	1.9	1.9	5.5	6.0
Total inorganic phosphorus, µg/l	27.5	6.0	5.0	8.0	8.0
NO_2-N, µg/l	0.7	0	0	0	0
NO_3-N, µg/l	10.0	2.0	2.6	4.0	4.0
Total N, mg/l	0.81	0.70	0.58	0.78	0.73
Biological oxygen demand, mg O_2/l	1.3	1.3	1.1	1.7	2.5
Total organic carbon, mg/l	11.6	11.1	14.0	12.2	11.9
Total suspended particulate carbon, mg/l	0.6	0.3	nd	1.1	1.2

SAMPLING PROCEDURE FOR POND MICROBIOTA

Pond microbiota sampled included bacteria, protozoa, algae, and zooplankton. During the 1-month study, samples were taken 5 times from each pond (except the control pond, where samples were taken 4 times). We sampled once prior to treatment and at 4-day intervals subsequent to dosing. Sediments were collected from each pond and subsampled for chlorophyll *a*, heterotrophic activity, and percentage organic matter.

Bacterial density was determined using epifluorescence microscopy and direct counts on membrane filters stained with erythrosin stain. Heterotrophic flagellates were quantified using epifluorescence microscopy with primulin stain. Rotifera were live-counted in Sedgewick-Rafter cells. Zooplankton were sampled using a 10-liter bucket following standardized techniques for small, shallow water bodies (Metodika izutchenia 1976). In each pond, ten 10-liter zooplankton samples were collected along a transect from the shore to the center for a total of 100 L water per sample. Each 100-liter sample was concentrated using a Juday net with brass screen (No. 73) to volume of 30 ml and preserved in 4 % formalin. In the laboratory, two subsamples of either 0.5, 1, or 2 ml (depending on visual density of zooplankters) were pipetted into Bogorov's counting chamber and zooplankters counted and identified using a binocular microscope.

To collect water mites, the water volume filtered for sampling was approximately 4.25 m^3 per sample. In all experimental ponds, except in the control, 3 samples were collected during the study period: before dosing, after 7 days, and after 11 days. In the control pond, the samples were collected before dosing and after 11 days.

Collected water mites were preserved immediately with Oudemans fixation liquid. The identification of water mites' taxonomic status was conducted in accordance with the classification of B.A. Wainstein (1980). Water mite species were identified using I.I. Sokolov's keyword book (1940).

DATA ANALYSIS

For species identification, we used Manuilova (1964), Kutikova (1970), Bening (1941) and Opredelitel presnovodnykh bespozvonotcnhykh (1977). For quantifying uncommon zooplankton species, the entire sample was enumerated. All zooplankters were divided into five groups: Cladocera, Copepoda, Rotatoria, copepod nauplii and copepod copepodites, and numerical analyses were performed on these groups.

The pond treatments were unreplicated. Comparisons were drawn from responses over time for the various taxonomic groups. Data were transformed by logarithms or arc-sine, as appropriate, to minimize correlations between the mean and variance of samples.

RESULTS

Before the permethrin application, planktonic communities differed among the ponds (Tables 2-4). At this time, Cladocerans dominated the zooplankton in all ponds studied; however, density of Cladocera, as well

Table 2. Density of common cladoceran species (individuals per liter pond water).

SPECIES	Density				
	sample day[1]				
	0	1	6	10	15
<u>**Control Pond**</u>					
Bosmina longirostris	3.6	3.6	12.5	-	31.2
Ceriodaphnia sp.	144.3	20.4	42.5	-	np[1]
Diaphanosoma brachiurum	0.6	3.0	7.0	-	19.0
Polyphemus pediculus	304.8	558.0	6.0	-	np
Simocephalus vetulus	6.3	1.2	14.5	-	np
Total density	465.0	600.0	98.0	-	63.6
Number of species	10	9	9	-	6
<u>**Acetone Control**</u>					
Bosmina longirostris	172.5	40.0	3.5	2.6	161.5
Ceriodaphnia sp.	139.6	3.7	3.0	105.9	26.0
Polyphemus pediculus	3.5	2.7	2.6	0.1	4.0
Scapholeberis mucronata	4.0	0.6	np	19.7	36.5
Total density	423.5	54.3	13.2	134.3	238.5
Number of species	10	10	7	9	9
<u>**0.5 µg/l**</u>					
Bosmina longirostris	0.3	43.5	101.5	0.5	3.0
Ceriodaphnia sp.	30.2	275.5	2.5	40.0	2.0
Polyphemus pediculus	40.3	8.0	36.6	14.1	3.0
Simocephalus vetulus	0.3	13.5	np	6.0	9.5
Total density	83.3	367.0	142.8	66.6	45.0
Number of species	11	9	6	9	12
<u>**5.0 µg/l**</u>					
Bosmina longirostris	np	0.9	112.0	42.0	69.0
Ceriodaphnia sp.	110.4	0.1	2.5	3.0	2.0
Diaphanosoma brachiurum	18.5	np	np	np	np
Polyphemus pediculus	6.0	np	3.5	np	7.5
Sida crystallina	4.8	np	np	1.0	0.5
Total density	144.3	1.0	122.5	47.3	89.5
Number of species	9	7	6	6	5
<u>**50.0 µg/l**</u>					
Bosmina longirostris	54.9	1.2	7.0	2.0	15.6
Ceriodaphnia sp.	307.8	np	np	np	np
Polyphemus pediculus	190.2	4.2	np	np	np
Total density	655.8	6.6	7.0	2.0	15.6
Number of species	10	4	1	1	1

[1]Days following pond dosing.
[2]np = not present.

Table 3. Density (individuals per liter pond water) of common copepod species and of nauplii and copepodites.

SPECIES	Density sample day[1] 0	1	6	10	15
Control Pond					
Cyclops sp.	0.6	0.6	np[2]	-	9.6
Eudiptomus gracilis	0.6	np	0.1	-	np
Heterocope appendiculata	np	1.8	0.1	-	np
Mesocyclops leuckarti	0.2	1.2	1.5	-	14.4
Paracyclops fimbriatus	np	np	np	-	7.2
Nauplii	np	np	7.0		464.4
Copepodites	0.1	6.0	15.5	-	538.8
Total density	1.8	3.6	1.7	-	32.4
Number of species	8	3	3	-	4
Acetone Control					
Acanthocyclops viridis	7.0	0.3	0.1	np	np
Eudiaptomus gracilis	3.6	1.2	0.8	np	np
E. graciloides	np	10.5	1.0	21.0	np
Heterocope apendiculata	4.0	6.3	0.3	7.0	np
Mesocyclops leuckarti	4.0	np	0.1	0.5	2.5
Nauplii	np	0.5	4.0	3.0	121.0
Copepodites	7.4	2.4	5.0	1.5	57.0
Total density	26.7	18.3	2.3	28.5	2.5
Number of species	5	4	5	3	1
0.5 µg/l					
Eudiaptomus gracilis	1.7	2.0	0.2	0.2	np
E. graciloides	np	1.0	np	np	np
Heterocope appendiculata	0.3	0.2	2.0	0.6	np
Mesocyclops leuckarti	np	2.0	np	3.5	10.0
Nauplii	np	4.5	12.0	14.5	8.0
Copepodites	1.2	1.1	18.0	11.5	14.0
Total density	2.4	5.4	2.2	4.4	10.0
Number of species	3	7	2	4	1
5.0 µg/l					
Eudiaptomus graciloides	np	0.6	1.0	0.3	np
Heterocope	0.3	1.0	1.5	0.1	np
Mesocyclops leuckarti	np	5.5	3.0	6.3	36.5
Nauplii	0.6	7.0	13.0	29.0	45.0
Copepodites	11.4	5.5	12.5	18.0	84.5
Total density	20.0	8.2	5.5	7.6	40.0
Number of species	5	6	3	7	4
50.0 µg/l					
Acanthocyclops viridis	3.0	nd	nd	nd	nd
Cyclops strenuus	13.2	nd	nd	nd	nd
Mesocyclops leuckarti	nd	nd	nd	u[3]	9.6
Nauplii	nd	2.4	1.0	3.0	9.6
Copepodites	9.6	0.6	1.0	3.0	39.6
Total density	16.2	0	0	0	9.6
Number of species	3	0	0	1	1

[1]Days following pond dosing.
[2]np = not present.
[3]u = single organism

Table 4. Density (individuals per liter pond water) of common Rotifera species.

SPECIES	Density				
	sample day[1]				
	0	1	6	10	15
Control Pond					
Asplanchna priodonta	np	6.0	10.0	-	12.0
Brachiounus angularis	np	np	35.0	-	200.4
B. calcyciflorus	np	0.6	13.0	-	1.2
B. diversicornis	np	np	5.5	-	202.8
Filinia longiseta	np	3.6	5.5	-	194.4
Keratella quadrata	np	4.2	21.0	-	1.2
Synchaeta sp.	0.3	np	np	-	np
Trichocerca cylindrica	np	np	4.5	-	327.6
Total density	0.3	18.6	169.0	-	1348.2
Number of species	1	8	19		18
Acetone Control					
Brachiounus angularis	1.0	1.5	1.0	np	32.0
B. calcyciflorus	1.0	2.0	1.0	1.0	np
Euchlanis dilatata	np	0.5	np	1.5	20.5
F. longiseta	np	np	1.0	np	35.0
Filinia mayer	4.0	2.0	1.5	2.0	np
K. cochlearis	2.0	2.0	0.5	3.0	1.0
Keratella quadrata	4.0	7.0	11.5	11.5	1.5
Total density	14.0	17.5	20.0	23.5	117.5
Number of species	7	8	11	10	11
0.5 µg/l					
Asplanchna priodonta	0.3	1.0	79.0	1.5	0.5
Euchlanis dilatata	np	np	np	8.0	8.0
Filinia longiseta	np	4.0	27.0	np	1.0
F. mayer	np	np	190.0	5.0	np
K. cochlearis	np	np	107.5	2.5	np
Keratella quadrata	0.3	6.5	342.0	53.0	3.0
Lecane luna	np	np	3.0	11.5	5.0
Total density	1.2	16.5	1008.5	94.0	21.0
Number of species	4	7	18	15	11
5.0 µg/l					
Asplanchna priodonta	1.5	2.5	71.0	44.0	np
Brachiounus calcyciflorus	np	6.5	66.0	157.0	np
Conochilus sp.	0.2	33.0	1.5	2.0	np
Euchlanis dilatata	nd	1.5	nd	64.0	23.0
Keratella quadrata	1.5	19.0	215.5	120.0	18.5
Total density	2.7	80.0	603.0	592.0	64.5
Number of species	3	15	21	21	11

Table 4. (continued)

SPECIES	Density sample day[1]				
	0	1	6	10	15
50.0 µg/l					
Asplanchna priodonta	226.8	139.2	30.0	8.0	48.0
Brachiounus calcyciflorus	np	0.6	66.5	9.0	32.4
Euchlanis dilatata	np	1.8	2.5	8.0	99.6
F. longiseta	np	np	18.0	6.0	296.4
Filinia mayer	np	np	np	35.0	np
K. cochlearis	np	np	11.0	28.0	12.0
Keratella quadrata	2.7	0.6	87.5	50.0	9.6
Lecane luna	2.4	np	4.0	6.0	164.5
Total density	236.1	144.0	283.5	206.3	786.7
Number of species	5	7	18	18	17

[1]Days following pond dosing.
[2]np = not present.

as dominant species from other orders, differed among ponds (Figure 1, Table 2). The lowest initial density of Cladocera was in pond 3 (83.3 individuals per liter), while the highest density was in pond 5 (655.8 individuals per liter). *Polyphemus pediculus* and *Ceriodaphnia* sp. dominated populations in ponds 1,3, and 5. *Bosmina longirostris* and *Ceriodaphnia* sp. Were dominant in pond 2, and *Ceriodaphnia* sp. and *Diaphanosoma brachiurum* in pond 4 (Table 2).

Initial copepod density was much less than that of the Cladocera (range 1.8 to 26.7 individuals per liter, Table 3). Copepod density was greatest in pond 2 and least in pond 1 (Figure 2, Table 3). Before treatment, copepod nauplii were found only in pond 4 and their density was very low, less than 1 nauplius per liter. Copepod copepodites were present in all ponds, but at very low densities (Table 3).

Initially, the highest rotifer density was in pond 5 (Table 4). However, this stemmed from one dominant rotifer species, *Asplanchna priodonta*, which, at 226.8 individuals per liter, was very high compared to the densities of all other rotifers in all ponds. Typically, total rotifer density was much lower, not exceeding 14.0 individuals per liter. Among the other rotifers, *Keratella quadrata* was dominant in ponds 2-4, while in the control pond, the only species present at this time was *Synchaeta* sp.

Density dynamics of zooplankton during the post-treatment period are shown in Figures 1-4. Quantitative and qualitative changes occurred in all ponds. By the end of the second week, cladoceran densities were reduced in all ponds; however, their response dynamics were different (Figure 1). In the control and 0.5 ug/l permethrin ponds, cladoceran

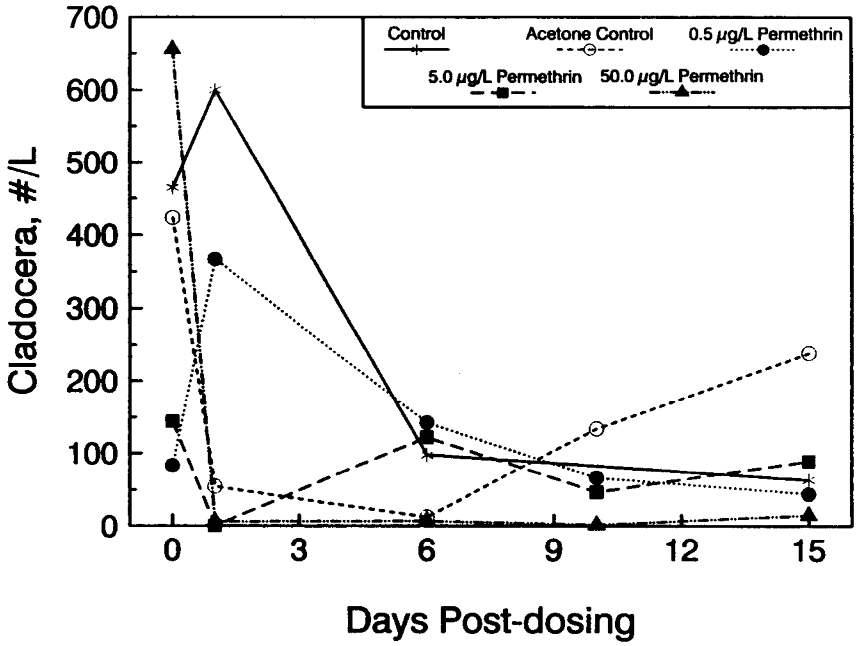

Figure 1. Number of cladocera per liter in IBIW experimental ponds dosed with permethrin.

density initially increased, gradually decreasing to less than pre-treatment densities. In these ponds, the initially codominant species, *Ceriodaphnia* sp. and *Polyphemus pediculus*, were replaced by *Bosmina longirostris* (in the control pond) and *Simocephalus vetulus* (in the 0.5 ug/l pond).

In ponds treated with 5.0 and 50.0 ug liter^{-1} permethrin (ponds 4 and 5, respectively), and in the control plus acetone pond (pond 2), cladoceran densities decreased dramatically beginning on day 1. After day 3, however, cladoceran density increased in ponds 2 and 4, but not in 5. The only cladoceran species that survived in pond 5 was *Bosmina longirostris*. This species became the most common cladoceran at the end of the observation period in the acetone control and the 5.0 ug liter^{-1} permethrin ponds (Table 1).

Density dynamics of early life-stage copepods were generally similar among the ponds (Figure 4). However, there were marked numerical differences among the populations. In the control pond, copepod nauplii and copepodites reached densities as high as 464.4 and 538.8 individuals per liter, respectively (Figures 3 and 4 respectively). In all other ponds, and especially in the 50.0 ug liter^{-1} permethrin pond, their densities were much less (Table 3). However, after day 15, there was a distinct increase in copepodites and nauplii. This phenomenon has been described by Solomon *et al.* (1980) and Kaushik *et al.* (1985). The copepodites and nauplii may show recovery as the permethrin concentrations in the ponds decrease.

By the end of the observation period, *Mesocyclops leuckarti* was the most common copepod species in all ponds. However, it was the only copepod species in the acetone control, 0.5 ug liter^{-1}, and 50.0 ug liter^{-1} permethrin ponds, while in the control and 5.0 ug liter^{-1} permethrin ponds, each copepod population consisted of four species. But, in fact, the copepod community was less complex in the 5 ug liter^{-1}, because *M. Leuckarti* was accompanied only by single individuals of *Cyclops* sp., *Acanthocyclops* sp., and *Ectocyclops* sp., while in the control pond *Cyclops* sp. and *Paracyclops fimbriatus* were codominant with *M. leuckarti* (Table 3).

Rotifers increased in both the control and acetone control ponds (Figure 5); however, there was an order of magnitude difference in density between the two ponds at the end of the experiment (Table 4). In the control pond, four rotifer species were codominant: *Filinia longiseta, Brachionus angularis, M. diversicornis* and *Trichocerca cylindrica*, while in the acetone control pond, *F. Longiseta, B.* angularis, and *Euchlanis dilatata* dominated the population.

Rotifer density in the 0.5 ug liter^{-1} permethrin ponds peaked 4 days after permethrin application and then decreased to pre-treatment

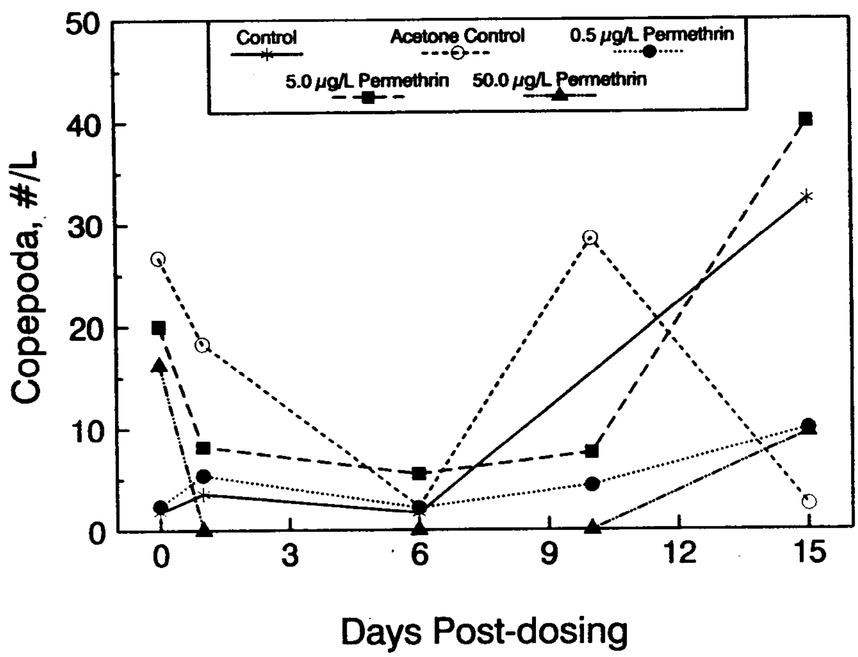

Figure 2. Number of copepods per liter in IBIW experimental ponds dosed with permethrin.

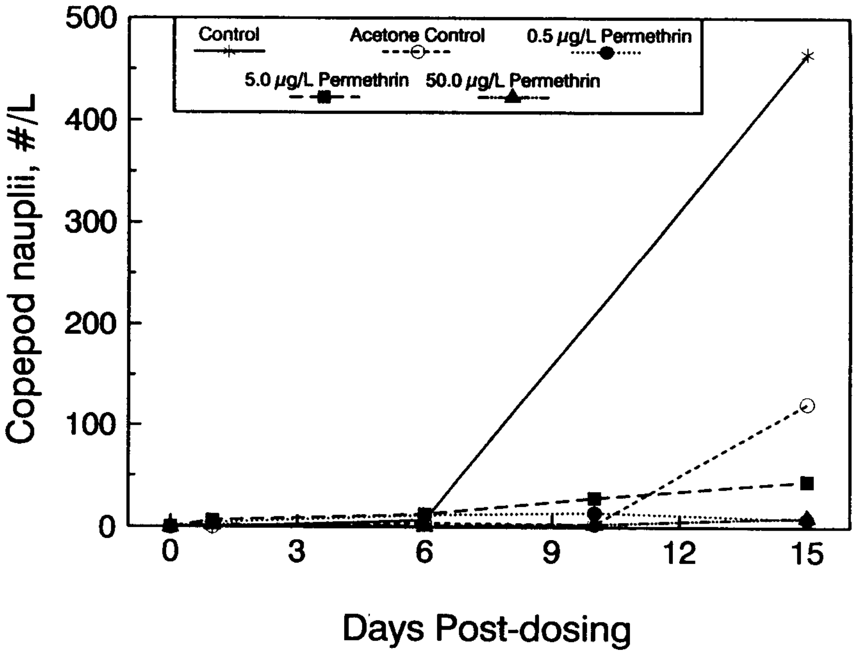

Figure 3. Number of copepod nauplii per liter in IBIW experimental ponds dosed with permethrin.

levels. The peaks in density were primarily a function of the increased number of *Keratella quadrata* in both ponds, *K.* cochlearia and *Filinia mayer* in pond 3, and *Asplanchna priodonta* and *Brachionas calyciflorus* in pond 4 (Figure 4). By the end of the experiment, *K. quadrata* and *Euchlanis dilatata* codominated the rotifer populations in both ponds, but their densities were low. In pond 5, 1 day after permethrin was applied, total rotifer density was slightly decreased. However, the rotifers gradually increased and reached a density of 786.7 individuals per liter. The dominant species at the beginning of the experiment in pond 5, *Asplanchna priodonta*, became much less abundant than *Filina longiseta* and *Euchlanis dilatata* by day 15. Interestingly, the rotifer population in pond 5 consisted of 17 species, which was nearly that of the control pond (with 18 species) (Table 4).

Water mites of *Eylais* and *Piona* genera were dominant in all the experimental ponds. The list of water mites collected from the IBIW ponds is provided in Table 5. Water mites of other species were found only in single exemplars. Before permethrin was added to any ponds, the greatest density of water mites of *Eylais* was found in pond 3 and for *Piona*, the greatest density was in pond 4. During the experimental period, the number of *Eylais* increased in ponds 1 (control) and 2 (control plus acetone) by 26% and 300%, respectively.

After permethrin was added, the number of water mites of both genera decreased in all treated ponds (except for the *Piona* mites in pond 3). In pond 3, after 7 days, the number of *Eylais* decreased 32.5% compared to pre-treatment levels and decreased by 29.1% after 11 days post-treatment. In pond 4, density decreased to 20.4% of original at 7 days and thereafter. The *Piona* in pond 3 increased in number, but this increase was very slight. We assume these mite densities, in fact, did not change after adding permethrin at a concentration of 0.5 ug liter^{-1}. In pond 4, the decline in number of water mites of this genus was much more pronounced. Seven days after treatment, the number was only 27.5%; 11 days after treatment only 14.5% compared to pre-treatment period. In pond 5, no water mites of either *Eylais* and *Piona*, as well as other genera, were found after pond dosing.

DISCUSSION

Published laboratory and field experiments have demonstrated that synthetic pyrethroids exert a toxic effect on zooplankton, especially crustaceans (Day 1989, Crossland 1982, Kaushik *et al.* 1984, Perevoznikov *et al.* 1984). Permethrin concentrations as low as 1.0 ug liter^{-1} and less were referred as acutely toxic for the variety of crustaceans (Day 1989). However, exposures simulating natural conditions help to complete a full evaluation of the effects of pyrethroid insecticides on natural planktonic communities.

Our pond experiments have shown that at a nominal permethrin concentration of 0.5 ug liter^{-1}, acutely toxic for a variety of aquatic invertebrates when tested in laboratory conditions, did not produce a marked direct effect even on species shown to be sensitive to a variety of toxicants, such as *Ceriodaphnia* (Winner 1988). Furthermore, while

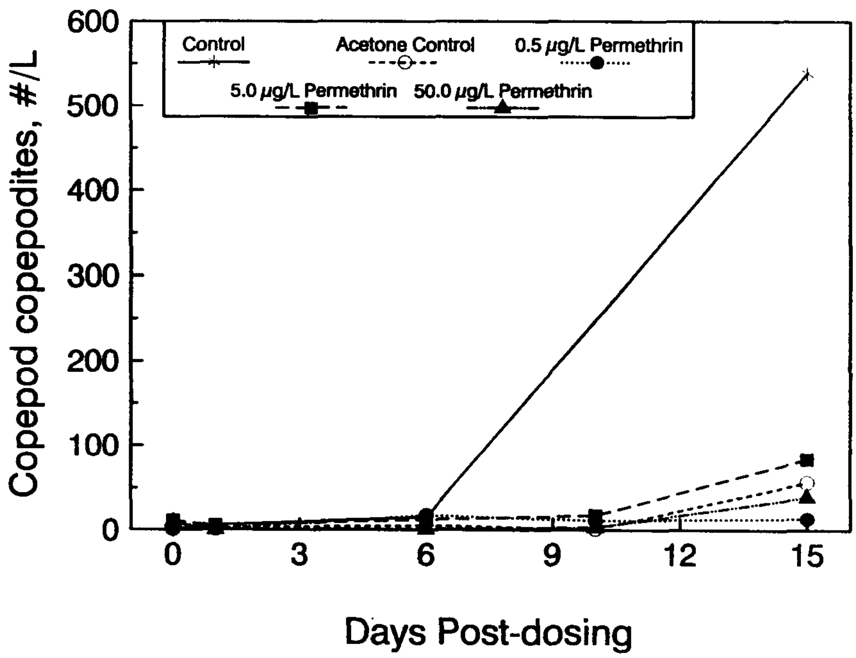

Figure 4. Number of copepod copepodites per liter in IBIW experimental ponds dosed with permethrin.

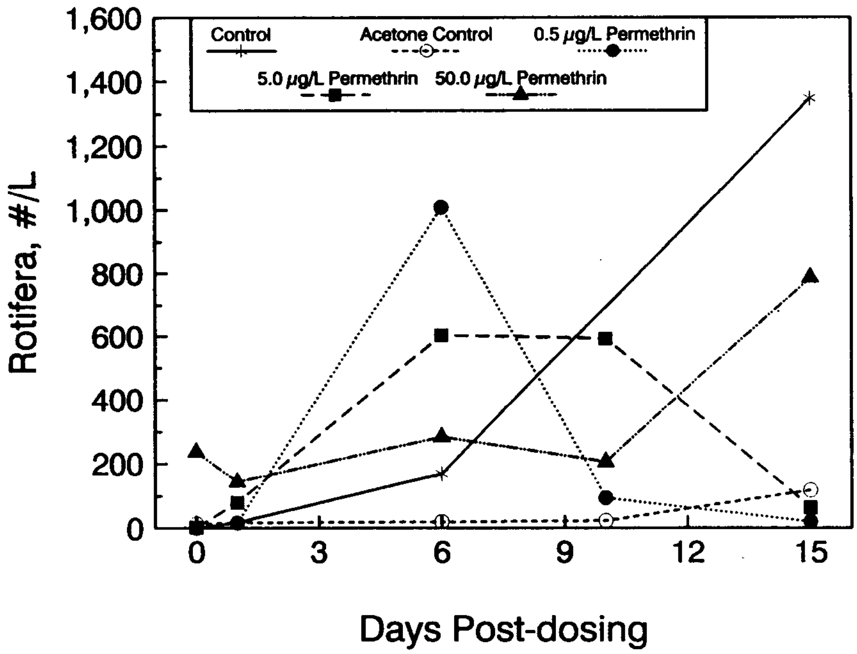

Figure 5. Number of rotifers per liter in IBIW experimental ponds dosed with permethrin.

5.0 ug liter^{-1} permethrin led to a dramatic decrease in density of cladocerans and copepods, the effects were less than would be expected on the basis of laboratory-measured responses.

Our results indicate that, in the control and acetone-control ponds, the density of *Eylais* water mites increased during the observation period. In the permethrin-treated ponds at concentrations of 0.5 and 5.0 ug liter^{-1}, their density decreased (Figure 1). On the other hand, *Piona* density decreased both in controls and in the 5.0 ug liter^{-1} permethrin-treated pond; however, in the treated pond, the effect was more pronounced. The concentration of 0.5 ug liter^{-1} seems to have no effect on this genus of mites (Figure 2). *Eylais* mites seem to be more susceptible to the permethrin than the *Piona*. However, at the permethrin concentration of 50.0 ug liter^{-1}, all *Eylais* and *Piona* water mites suffered complete mortality.

Hence, results of this study demonstrate the susceptibility of certain water mites to permethrin under field conditions. This is in good accordance with published results on the toxicity of insecticides to water mites (Nair *et al.* Stephenson *et al.* 1986, Crossland and Wolff 1985). However, our results are quite different from the results of investigations on the effects of industrial pollutants on aquatic mites (Alexeev 1986). Evidence exists to indicate that the primary reason for the elimination of water mites was the direct toxic effect of permethrin. That the elimination of this planktonic invertebrate was due to an elimination of more sensitive prey zooplankters seems unlikely because the mites have been shown to survive more than 15 to 20 days when starved (Zhavoronkova, unpubl.).

Table 5. Acari present in Sunuga ponds.

FAMILY	SPECIES
Hydrachnidae	*Hydrachna sp.*
Eylaidae	*Eylais mulleri Koenike, 1897; E. extendens O.F. Mull., 1776; E. hamata Koenike, 1897; E. koenikei, 1903*
Hydriphantidae	*Hydryphantes sp., Hydrodroma sp.*
Limnesiidae	*Limnesia sp.*
Hygrobatidae	*Piona coccinea (C.L. Koch), 1836; P. nodata (O.F. Muller), P. variabilis (C.L. Koch), 1836; Hydrochoreutes sp., Acercus sp.*
Arrhenuridae	*Arrhenurus sp.*

Our experiment was performed in mature ponds, all with extensive macrophyte growth. The biomass of submerged higher aquatic plants was high, as was the concentration of suspended solids. The propensity of permethrin to adsorb quickly onto solid surfaces (e.g., suspended solids, particulate organic matter, etc.) is well known (Muir *et al.*, Sharom and Solomon 1979).

The IBIW ponds are relatively productive. High ecosystem productivity, with associated high plant surface area and concentration of particulate organic material, lessened the bioavailability of pentachlorophenol in a mesocosm study (Robinson-Wilson *et al.* 1983). Functional redundancy in the macrophyte community within mesocosms also lessens the potential for measuring ecological damage (Fairchild *et al.* 1994). Empirical laboratory results (McCarthy and Bartell 1988) show that for periphyton, macrophytes and detritivorous fish, there is a decrease in toxic effects of phenol to daphnia. The lessened toxicity was directly related to increased particulate organic matter (POM). However, for phytoplankton , zooplankton, benthic invertebrates and omnivorous fish, greater reduction in toxic effects corresponded to increasing dissolved organic material (DOM) in the system. Bacterial and zooplanktonic growth was dramatically enhanced as a function of available organic material and, presumably, greater surface area for microbial activity.

Higher levels of primary production corresponded to lower pentacholorophenol (PCP) residues and toxicity to fish in a pond-mesocosm study (Robinson-Wilson *et al.* 1983). Yet, lower PCP toxicity could not be ascribed solely to an increase in primary production. Their results indicated that the larger surface area available within the macrophyte-dominated ponds contributed to an active epiphytic community which degraded the PCP. In a similar manner, the adsorption of permethrin onto plant surfaces and suspended solids in these ponds was apparently high and may explain the less pronounced direct toxic effect of the 0.5 and 5.0 ug liter^{-1}, permethrin concentrations, and the greater susceptibility of active filter-feeding cladocerans (*Polyphemus pediculua, Sida crystallina, Diaphanosoma brachiurum*) and copepods (*Eudiaptomus gracioloides*), while predatory copepods such as *Mesocyclops leuckarti* appeared to be much less susceptible to the insecticide. Among the active filter-feeding zooplankters, such as *Polyphemus pediculus, Daphnia longispina, Sida crystallina, Eudiaptomus gracilis,* and *E. Graciloides*, are the most sensitive to the toxic effects of permethrin. Of the zooplankton collected in our experiment, Rotifera were the most resistant to permethrin. This agrees well with previous results reported by Kaushik *et al.* (1985). The reasons for such resistance, as well as factors influencing rotifer population dynamics as a function of permethrin, were discussed by these authors in detail.

The reasons for the noted changes in zooplankton density following acetone application are not clear. The decline in Cladocera no doubt stems from unmeasured pond-to-pond differences. Such pond-to-pond differences have recently been documented in a separate study (Hellenbrant 1994). Different rates of decline and growth in zooplankton as a result of permethrin exposure were observed by Kaushik *et al.* (1985). They note zooplankton numerically respond to an insecticide in two ways, direct reduction from acute toxicity and indirect numerical responses attributed to release from predatory or competitive pressures.

There is evidence that the recovery of zooplankton populations in mesocosms is a function of their relatively short generation times, the number of times they are exposed, and the persistence of the pesticide in the water column (Fairchild *et al.* 1991). In that study, zooplankton responded much more dramatically to pulsed addition (6 dosing periods over 12 weeks) of esfenvalerate than to similar concentratioins dosed twice over the same period. There was a minimal effect on cladocerans in ponds dosed one time at 0.1 mg liter^{-1} carbaryl (Hanazato and Yasuno 1990). In ponds dosed twice, the cladocerans were able to recover within to 15 days. However, in ponds dosed 10 times over a period of 20 days, each time at the 0.1 mg liter^{-1} concentration, the cladocerans failed to recover.

Recovery of copepod densities after permethrin application in this experiment occurred after 2 weeks, even in the highest concentration. The major source of recovery was the appearance of the neonatal individuals (Figures 3 and 4). High resistance of eggs of these copepods to a variety of the environmental stressors has been noted by previous investigators (Makrushin 1984). It is important to note that another route of recovery for planktonic populations exists, that of immigration of smaller or younger forms into the experimental ponds. This remains an uncontrolled variable in similar ponds and mesocosm studies.

The effect of pesticides on aquatic organisms needs to be studied relative to the potential for recovery and recolonization of the species. An important aspect of disturbance on communities is the rate at which the habitat becomes less suitable in relation to the requirements of the organisms (Southwood 1988). In effect, the number of dosing intervals and the concentration of the pesticide become points on disturbance and adversity axes. Over sufficient time, disturbance and periods of adversity have a strong influence on the biotic structure of communities. Incorporating such factors into future mesocosm studies will allow a study of species dynamics relative to the degree and frequency of disturbances.

REFERENCES

Anderson, R.L. 1989. Toxicity of synthetic pyrethroids to freshwater invertebrates. Environmental Toxicology and Chemistry 8:403-410.

Associate Committee on Scientific Criteria for Envionmental Quality. 1986. Pyrethroids: their effects on aquatic and terrestrial ecosystems. NRCC PubL. No. 24376. National Research Council of Canada, Ottawa, Ontario.

Barnthouse, L.W., G.W. Suter II, A.E. Rosen and J.J. Beauchamp. 1987. Estimating responses of fish populations to toxic contaminants. Environmental Toxicology and Chemistry 6: 811-824.

Barnthouse, L.W., G.W. Suter II and A.E. Rosen. 1990. Risks of toxic contaminants to exploited fish populations: Influence of life history, data uncertainty and exploitation intensity. Environmental Toxicology and Chemistry 9: 297-311.

Bening A. L. 1941. Cladocera _ (Cladocera of the Caucasus region). Tbilisi,''is'' publ.house, 383 pp. (In Russian).

Cairns, J., Jr. 1983. Are single species toxicity tests alone adequate for estimating environmental hazard? Hydrobiologia 100: 47-57.

Coats, J.R. and S.P. Bradbury. 1989. Aquatic toxicology of synthetic pyrethroid insecticides. Environmental Toxicology and Chemistry 8:359.

Crossland, N.O. 1982. Toxicology of cypermethrin. II. Fate and biological effects in pond experiments. Aquatic Toxicology 2:205-SSS.

Crossland, N.O. and C.J.M. Wolff. 1985. Fate and biological effects of pentachlorophenol in outdoor ponds. Environmental Toxicology and Chemistry 4:73-86.

Day, K.G. 1989. Acute, chronic and sublethal effects of synthetic pyrethroids in freshwater zooplankton. Environmental Toxicology and Chemistry 8:411-418.

Day, K.E.and N.K. Kaushik. 1987. Short-term exposure of zooplankton to the synthetic pyrethroid, fenvalerate, and its effects on rates of filtration and assimilation of the alga, *Chlamydomonas reinhardii*. Archives of Environmental Contamination and Toxicology 16:423-432.

Day, K.E., N.K. Kaushik and K.R. Solomon. 1987. Impact of fenvalerate on enclosed freshwater plankton communities. Canadian Journal of Fisheries and Aquatic Sciences 44:1714-1728.

DeNoyelles, F., Jr., S.L. Dewey, D.G. Huggins and W.D. Kettle. 1994. Aquatic mesocosms in ecological effects testing: Detecting direct and indirect effects of pesticides. Pps. 577-603; In R.L. Graney, J.H. Kennedy and J.H. Rodgers, Jr. (Eds). Aquatic Mesocosm Studies in Ecological Risk Assessment. CRC Press, Inc. Boca Raton, FL.

Dewey, S.L. and F. DeNoyelles, Jr. 1994. On the use of ecosystem stability measurements in ecological effects testing. Pps. 605-625; In R.L. Graney, J.H. Kennedy and J.H. Rodgers, Jr. (Eds). Aquatic Mesocosm Studies in Ecological Risk Assessment. CRC Press, Inc. Boca Raton, FL.

Fairchild, J.F., T.W. La Point, J.L. Zajicek, M.N. Nelson and F.J. Dwyer. 1991. Population, community, and ecosystem-level responses of aquatic mesocosms to pulsed doses of a pyrethroid insectide. Environmental Toxicology and Chemistry 11:115-129.

Fairchild, J.F., T.W. La Point and T.R. Schwartz. 1994. Effects of an herbicide and insecticide mexture in aquatic mesocosms. Archives of Environmental Contamination and Toxicology 27:527-533.

Ferevaznikov M A. , A. A. Salaskin, A. V. Maksuchin, A. P. Lysak. 1988. (Ecological tolerance of Cladocerans and Copepods to pesticides) p.p. 42-53 in: M A. Cerevaznikov, A. N.._Gantverg (eds) (The influence of biologically active chemicals upon hydrobionts). Leningrad, ''od'' publications (In Russian).

Giddings, J.M., R.L. Helm and F.J. deNoyelles, Jr. 1994. Large-scale outdoor microcosms: Tools for ecological assessment of pesticides. Pps. 191-198; In I.R. Hill, F. Heimbach, P. Leeuwangh and P. Matthiessen (eds.) Freshwater Field Tests for Hazard Assessment of Chemicals. CRC Press, Inc. Boca Raton, FL.

Gledhill, T. 1985. Water mites - Predators and parasites. Freshwater Biology Association Annual Report 53:45-59.

Hamilton, P.B., G.S. Jackson, N.K. Kaushik, K.R. Solomon, and G.L. Stephenson. 1988. The impact of two applications of atrazine on the plankton communities of in situ enclosures. Aquatic Toxicology 13: 123-140.

Hanazato, T. and M. Yasuno. 1990. Influence of persistence period of an insecticide on recovery patterns of a zooplankton community in experimental ponds. Environmental Pollution 67: 109-122.

Hellenbrandt, S. 1994. A comparison of zooplankton sampling methods in evaluating copper sulphate toxicity in microcosms. Unpublished MS thesis; University of Aachen, Germany.

Johnson, R.K., T. Wiederholm and D.M. Rosenberg. 1993. Freshwater biomonitoring using individual organisms, populations, and species assemblages of benthic macroinvertebrates. In D.M. Rosenberg and V.H. Resh (eds). Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman & Hall, NY.

Kaiser, K.L.E. 1980. Correlation and prediction of metal toxicity to aquatic biota. Canadian Journal of Fisheries and Aquatic Sciences 37:211-218.

Kaushik, N.K., G.L. Stephenson, K.R. Solomon and K.E. Day. 1985. Impact of permethrin on zooplankton communities in limnocorrals. Canadian Journal of Fisheries and Aquatic Sciences 42:77-85.

Kimball, K.D. and S.A. Levin. 1985. Limitations of laboratory bioassays: the need for ecosystem-level testing. BioScience 35(3) 165-171.

Kutikova, L.A. 1970. Kolovratki fauny SSSR _(Rotatorias of the USSR fauna). Leningrad, ''Nauka"' publ. (In Russian).

Kutikova, K.A. 1977. (The key book for the determiustion of the freshwater invertebrates of the European part of the USSR). L.A. Kutikova' Ya. I. Stov, eds. Leningrad,''Gidrometxcizdat'' publ. house, (In Russian).

Kutrushin A.V. 1938. (The use of different Cladoceran species for biotesting) p. 92-96 in: Ferevaznikov M A., Gantveog A.N. (eds) The influence of biologically activites upon hydrobionts). Leningrad, publications 140 pp. (In Russian).

La Point, T.W. 1994. Interpreting the results of agricultural microcosm tests: Linking laboratory and experimental field results to predictions of effect in natural ecosystems. Pages 83-102, In I.R. Hill, F. Heimbach, P. Leeuwangh and P. Matthiessen (eds.) Freshwater Field Tests for Hazard Assessment of Chemicals. CRC Press, Inc. Boca Raton, FL.

La Point, T.W., J.F. Fairchild, E.E. Little, and S.E. Finger. 1989. Laboratory and field techniques in ecotoxicological research: Strengths and limitations. In A. Boudou and F. Ribeyre (eds.) Aquatic Ecotoxicology: Fundamental Concepts and Methodologies, Vol. II. CRC Press, Boca Raton, Florida. pp. 239-255.

Larsen, D.P., F. deNoyelles, F. Stay and T. Shiroyama. 1986. Comparisons of single-species, microcosm, and experimental pond responses to atrazine exposure. Environmental Toxicology and Chemistry 5:179-190.

Liber, K., K.R. Solomon, N.K. Kaushik and J.H. Carey. 1994. Impact of 2,3,4,6-tetrachlorophenol (DIATOX) on plankton communities in limnocorrals. Pps. 257-294; In R.L. Graney, J.H. Kennedy and J.H. Rodgers, Jr. (Eds). Aquatic Mesocosm Studies in Ecological Risk Assessment. CRC Press, Inc. Boca Raton, FL.

Manuilova E.F. 1984. (Cladocera of the USSR fauna). Leningrad,''Nauka"' publ. house, 989 pp. (In Russian).

McCarthy, J.F. and Bartell, S.M. 1988. How the trophic status of a community can alter the bioavailability and toxic effects of contaminants. In J. Cairns, Jr. and J.R. Pratt (eds.) Functional Testing of Aquatic Biota for Estimating Hazards of Chemicals. ASTM STP 988. American Society of Testing and Materials, Philadelphia, PA. pp. 3-16.

Methods of the investigation of the inland water biogeohydrobionts). 1976. F. O.i-Boltovskyup, ed. Leningrad, ''Nauka"' publ. house, 240 pp.(In Russian).

Minshall, G.W. 1984. Aquatic insect-substratum relationships. Chapter 12 In V.H. Resh and D.M. Rosenberg (eds). The Ecology of Aquatic Insects. Praeger, NY.

Muir, D.G.C., G.P Rawn and N.P. Grift. 1985. Fate of the pyrethroid insecticide deltamethrin in small ponds: a mass balance study. Journal of Agricultural and Food Chemistry 33:581-591.

Robinson-Wilson, E.F., T.P. Boyle, and J.D. Petty. 1983. Effects of increasing levels of primary production on pentachlorophenol redisues in experimental pond ecosystems. In W.E. Bishop, R.D. Cardwell, and B.B. Heidolph (eds.) Aquatic Toxicology and Hazard Assessment: Sixth Symposium. ASTM STP 802. American Society of Testing and Materials, Philadelphia, PA. pp. 23;9-251.

Solomon, K.R., K. Smith, G. Guest, J.Y. Yoo and N.K. Kaushik. 1980. Use of limnocorrals in studying the effects of pesticides in the aquatic ecosystem. Canadian Technical Reports on Fisheries and Aquatic Sciences 975:1-9.

Southwood, T.R.E. 1988. Tactics, strategies and templets. Oikos 52: 3-18.

Stephenson, G.L., N.K. Kaushik, K.R. Solomon and K. Day. 1986. Impact of methoxychlor on freshwater communities of plankton in limnocorrals. Environmental Toxicology and Chemistry 5:587-603.

Ten Winkel, E.H. and C. Davids. 1985. Bioturbation by cyprinid fish affecting the food availability for predatory water mites. Oecologia 67:218-219.

Wainstein, B.A. 1980. (Key to larval water mites) Institut Biologii Vnutrennikh Vod. (Academiia Nauk SSSR) 238 pp. (In Russian)

Winner R.W. 1938. Evaluation of the relative sensitivity of 7-d *Daphnia magna* and *Ceriodaphnia dubia* toxicity tests for cadmium and cadmium pentachlorophenate. Environmental Toxicology and Chemistry 7:153-159.

Yasuno, M., T. Hanazato, T. Iwakuma, K. Takamura, R. Ueno and N. Takamura. 1988. Effects of permethrin on phytoplankton and zooplankton in an enclosure ecosystem in a pond. Hydrobiologia 159:247-258.